Estuarine, Coastal and Shelf Science 223 (2019) 147–158

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Effect of introduced *Casuarina* trees on the vulnerability of sea turtle nesting beaches to erosion

Deidre de Vos[a,*], Ronel Nel[a], David Schoeman[c], Linda R. Harris[a], Derek du Preez[b]

[a] Institute for Coastal and Marine Research, Department of Zoology, P.O. Box 77000, Nelson Mandela Metropolitan University, Port Elizabeth, 6031, South Africa
[b] Department of Botany, P.O. Box 77000, Nelson Mandela Metropolitan University, Port Elizabeth, 6031, South Africa
[c] Faculty of Science, Health & Education, University of the Sunshine Coast, Maroochydore, Q-4558, Australia

ARTICLE INFO

Keywords:
Shore protection
Bio-shield
Backshore
Beach erosion
Integrated coastal management

ABSTRACT

The world's beaches are under severe pressure because of anthropogenic modification and climate change. The most common coastal management approaches have focused largely on the protection of infrastructure without considering the function of the beach and dune system as a natural protective barrier and a diverse ecosystem. Large-scale plantations of non-native vegetation have been initiated in many countries as a coastal protection tool. However, exotic trees commonly used for bio-shields, such as *Casuarina equisetifolia,* have been shown to have negative effects on the coastal environment, including the promotion of beach erosion in some instances and negative impacts on sea turtles. The aim of the study was to determine if exotic *Casuarina* spp. enhance the vulnerability of turtle nesting beaches by exacerbating erosion. The extent of use of exotic *Casuarina* spp. was assessed on 50 important sea turtle nesting beaches of the Indian Ocean and South-East Asia (IOSEA) region and used an adapted Coastal Vulnerability Index (CVI) to determine vulnerability of these beaches to erosion. Sites were assessed using long-term global data sets under four risk indicators (backshore width, beach exposure, modal beach energy, dune state) and three threat indicators (coastal development, sea-level rise and storminess). Beach vulnerability was then classified into four categories: 1) High Risk-High Threat; 2) High Risk-Low Threat; 3) Low Risk-High Threat and 4) Low Risk-Low Threat. Overall, *Casuarina* trees were present on 28% of beaches, and seem to be a popular management option in the IOSEA region. *Lepidochelys olivacea* rookeries were most affected by *Casuarina*, which is of concern because these populations are already considered at risk from other stressors. Overall, beaches tended to have higher risk than threat, and beaches with *Casuarina* present were not shown to be more vulnerable to erosion (Kruskal-Wallis $X^2_{44} = 42.62$, $p = 0.5308$). However, these trees may pose other threats to sea turtles, e.g., providing obstructions to nesting females and altering nest incubation environments, which need to be carefully considered by decision makers. Ideally, natural dune cordons need to be maintained and restored, conferring benefits of protection to people, and benefits of ecological processes to biodiversity.

1. Introduction

Rapid global change and anthropogenic modification of natural habitats are threatening ecosystems across the globe (Canadell et al., 2007; Halpern et al., 2015). Many of these pressures are concentrated on the world's coastlines because an increasingly large proportion of the global population is choosing to live next to the coast (Crowell et al., 2007). As a result, the dynamic littoral active zone is frequently stabilised with hard engineering structures, such as sea walls (Charlier et al., 2005; Bulleri and Chapman, 2010) to protect inappropriately located infrastructure from wave damage. However, these walls also disrupt coastal ecological processes by breaking land-sea connectivity. Given the negative consequences of hard stabilisation, e.g., coastal squeeze (Brown and McLachlan, 2002; Defeo et al., 2009), there is increasing use of 'soft' options like large-scale plantations of non-native bio-shields (Mukherjee et al., 2009; Tanaka, 2009; Samarakoon et al., 2013) to protect the hinterland but with fewer ecological impacts. Indeed, the popularity of green shelter belts has rapidly increased since the 2004 Indian Ocean tsunami. Consequently, extensive portions of coastal habitat have been modified by planting exotic vegetation, such as *Casuarina equisetifolia* (Danielsen et al., 2005; Tanaka, 2009; De Zoysa, 2008).

Despite their ubiquitous use, the functionality of *Casuarina* trees as a coastal protection tool has yet to be scientifically tested. Indeed, studies

* Corresponding author. Department of Zoology, PO Box 77000, Nelson Mandela University, Port Elizabeth, 6031, South Africa.
E-mail address: devos.deidre@gmail.com (D. de Vos).

https://doi.org/10.1016/j.ecss.2019.03.015
Received 28 September 2018; Received in revised form 4 March 2019; Accepted 24 March 2019
Available online 01 April 2019
0272-7714/ © 2019 Elsevier Ltd. All rights reserved.

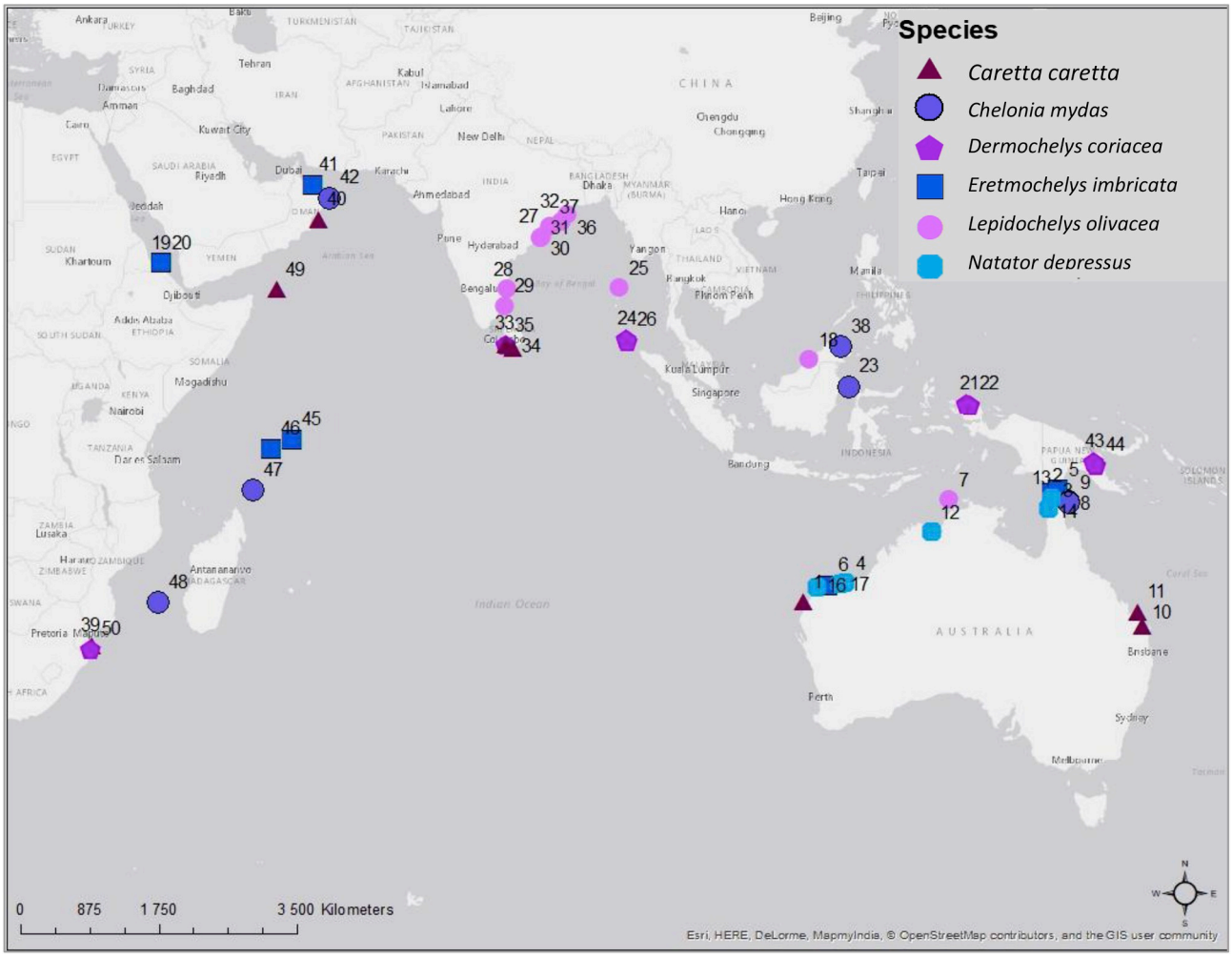

Fig. 1. a) Distribution of the 50 study sites (sea turtle nesting beaches). The colour and shape of the point represents the sea turtle species indicated in the map legend. The study sites are labelled with numbers corresponding to the site names in Appendix 1: Study sites/sea turtle nesting beaches. b) Distribution of the beaches used in the training data set published by Defeo and Mclachlan (2013) (n= > 200) to develop new risk/threat score categories. Refer to the online version of the article to see the figures in colour. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

have emerged disputing the sustainability of using non-native trees as dune stabilisers or storm-protection measures (Bhalla, 2007; Das and Sandhu, 2014), especially given their adverse effects on sandy beach ecosystems (Gordon, 1998; Wheeler et al., 2011). For example, after an initial period of relatively slow increase, *Casuarina* spp. increase exponentially (Potgieter et al., 2014a). They out-compete and replace native plant communities (Hardman et al., 2012), create a sterile soil environment (Jadhav and Gaynar, 1995) and alter local soil biogeochemistry (Batish et al., 2001; Buehler and Rodgers, 2012; Vitousek, 1990), thereby compromising growth of other plants (Patil et al., 2002), and reducing animal diversity (Mazzotti et al., 1981). Further, their root systems cannot retain sand (Sealey, 2006), which alters natural sediment dynamics across the littoral active zone, thereby increasing shoreline steepness (Chaudari et al., 2009). This leads to changes in other physical beach features that are indicators of erosion, e.g., back beach width (Rizzo et al., 2017) or the state of the dune system (Mclaughlin and Cooper, 2010).

The concern is that substantial portions of backshores globally now have non-native *Casuarina* species present, including sea turtle rookeries. In India, for example, an estimated 180 000 *C. equisetifolia* saplings have been planted on 450 ha of coastal habitat (Chaudari et al., 2009) and some of these plantations back olive ridley (*Lepidochelys olivacea*) nesting beaches (Chaudari et al., 2009). In fact, almost a third of India's coastline has been covered with plantations since extensive coastal planting programmes were initiated in response to the 2004 Indian Ocean tsunami (Danielsen et al., 2005; Chaudari et al., 2009; Mukherjee et al., 2009). Similar reports of dune stabilisation using *Casuarina* trees are found in China (Zhong et al., 2010), Sri Lanka (Mathiventhan and Jayasingam, 2014), USA (Klukas, 1969), and South Africa (Potgieter et al., 2014b), some of which back sea turtle rookeries. The risk of these trees to turtles is both indirect by increasing erosion of the nesting beaches (as described above), and direct by altering nest incubation environments (Schmelz and Mezich, 1988; Schmid et al., 2008; Chaudari et al., 2009) and creating obstructions to nesting females on the back beach if the trees topple over in storms (Klukas, 1969).

Casuarina trees could thus be a notable but overlooked threat to sea turtles that consequently needs to be quantified. Given the prevalence of these introduced trees along shores bounded by the Indian Ocean, which beaches also support nesting turtles, this region serves as an excellent study area to investigate this threat. Further, at the 7th convention of the Signatory States of the Indian Ocean and South-East Asian Marine Turtle Memorandum of Understanding (IOSEA MoU), the development of large-scale coastal bio-shields, often with exotic vegetation, was raised as a red flag. One tool to assess erosion vulnerability of sea turtle nesting beaches is a Vulnerability Assessment (VA). VAs highlight the balance between risks and threats and so identify areas susceptible to damage (UNDRO, 1982). Coastal managers can use VAs

to identify regions at risk where existing stressors (risks) might be exacerbated by global-change threats (Anfuso and Martínez Del Pozo, 2009, Mclaughlin and Cooper, 2010). Alternatively, VAs could be used to inform suitable risk-management priorities or planning strategies (Glick et al., 2011).

The aim of this study is to estimate the erosion vulnerability of important sea turtle nesting beaches in the Indian Ocean and South East Asia (IOSEA) region to determine whether *Casuarina* trees enhance erosion vulnerability over and above other factors, such as urban development. The specific objectives are 1) to quantify the distribution of non-native *Casuarina* trees on sea turtle nesting beaches of the IOSEA region and 2) to create an erosion vulnerability score (based on risk and threat) for the nesting sites based on global datasets of erosion indicators. Then, 3) to apply the vulnerability index to 50 sea turtle nesting beaches, and 4) to assess whether the presence of *Casuarina* trees could be used as an indicator of enhanced beach erosion. We hypothesise that the characteristics of *Casuarina* trees are not conducive to them being an effective coastal protection tool and predict that beaches backed by these trees will have higher erosion vulnerability scores than those beaches without these trees.

2. Materials and methods

2.1. Study sites

The IOSEA region consists of the 35 Signatory States to the IOSEA Sea Turtle MoU (Map of the IOSEA signatory states: http://www.ioseaturtles.org/org_map.php). Within this region, 50 sea turtle rookeries were selected (Fig. 1a). For each of the six turtle species present in the Indian Ocean, beaches with the highest nesting densities (hereafter considered: important nesting beaches) were selected as reported in the State of the World's Sea Turtles (SWOT) database (http://seamap.env.duke.edu/swot). Some beaches host more than one species; if a beach selected for one species was already selected as important for another species, then an additional beach was chosen for the former until a total of 50 beaches were selected. This allowed for a balanced distribution of nest sites among turtle species whilst also having a representative selection of beaches across the region and among countries. Selected study sites included eight *Caretta caretta*, seven *Chelonia mydas*, eight *Dermochelys coriacea*, nine *Eretmochelys imbricata*, eleven *Lepidochelys olivacea* and seven *Natator depressus* nesting beaches (Fig. 1a; SM: Sea turtle nesting beaches). We recognise that, although Australia is a Signatory State to the IOSEA Marine Turtle MOU, the beaches on the east coast are not IOSEA beaches. Nevertheless, they are important rookeries for this Signatory State, and thus were included in the study. For simplicity, we still refer to the study area as the IOSEA region.

2.2. Casuarina distribution

Casuarina cunninghamiana, *Casuarina equisetifolia* and *Casuarina glauca* are the only *Casuarina* species globally recorded as either naturalised or invasive in coastal regions (Potgieter et al., 2014a) and were therefore the focus of this study. To establish their occurrence on turtle rookeries across the IOSEA region, several approaches on different scales were considered. First, the global distribution and native ranges were established using three global data sets: The Invasive Species Compendium (CAB International, 2000; http://www.cabi.org/ISC/), Global Biodiversity Information Facility (GBIF 2008; http://www.gbif.org) and Atlas of Living Australia (http://www.ala.org.au/). These datasets, however, do not allow for identification at a local scale (i.e. beach level). To establish *Casuarina* tree occurrence per beach, user-posted images on Google Earth (Pro 7.3.0.3832) Panoromio were used. Geotagged images uploaded by users allowed us to identify *Casuarina* presence and extent along the backshore, but not the species. Only native vs non-native *Casuarina* presence and absence information was indicated with no assumptions about the species or impact on each of the different nesting beaches. Beaches were therefore defined according to *Casuarina* presence and origin. The purpose of indicating non-native *Casuarina* presence was to inform analysis of beach erosion vulnerability in the presence and absence of *Casuarina* trees.

2.3. Vulnerability assessment: the Coastal Vulnerability Index (CVI)

2.3.1. Risk and threat indicators

To calculate beach erosion vulnerability as a measure of *risk* and *threat*, the Coastal Vulnerability Index (CVI) developed by Gornitz et al. (1994) for site-specific studies was adapted for use at a regional scale. Risk indicators, as suggested by existing literature, evaluate physical beach features, including: backshore width (Rizzo et al., 2017); beach exposure (Bryan et al., 2001; Goodhue et al., 2012); modal beach energy, as measured by wave height and tide range (Gornitz et al., 1994, Mclaughlin et al., 2002; Abuodha, 2006); and state of the dune system (Mclaughlin and Cooper, 2010). Threat indicators focus on erosion threats to beaches, such as: development (Benassai et al., 2015; Rizzo et al., 2017); sea-level rise (Pethick, 2001; Corbella and Stretch, 2012); and storm frequency and intensity (Gornitz et al., 1994; Mclaughlin and Cooper, 2010; Li and Li, 2011). The indicators included in this study included those above, modified by data availability at an ocean-basin scale.

2.3.2. Training data set

Where scoring categories per indicator were either not available from the literature or not applicable at a regional scale, new categories were developed to score the indicators. To define these categories, a beach database published by Defeo and Mclachlan (2013) was used as a training data set (Fig. 1b) because it contains comprehensive information for a representative sample ($n > 200$) of beaches, globally. Values for the indicators were obtained for beaches in the training data set and the distribution of the values plotted, with the 0–20, 20–40, 40–60, 60–80 and 80–100 percentiles providing the score categories per indicator. Indicator values measured for beaches in the training data set were therefore taken as the null distribution against which corresponding scores for the turtle-nesting beaches could be compared. Wherever insufficient data were available, the indicator was scored as data deficient and, following Wallace et al. (2011), was assigned the highest score for that indicator.

2.3.3. Beach vulnerability

To avoid subjectivity in our assessment, no weights were applied to the indicators (Mclaughlin and Cooper, 2010; Benassai et al., 2015). Scores were normalised for each indicator to yield a value between zero and one (i.e. allocated score was divided by the maximum score of that category) and beach vulnerability was visualised by plotting the sum of normalised scores for risk indicators (x-axis) against the sum of normalised scores for threat indicators (y-axis). Recognising that it might be impossible for a beach to attain the highest score in every category (i.e., have no backshore, experience rapid sea-level rise, and very frequent extreme storms) and that scoring the lowest value for every category gave a total that was not zero, we calculated the theoretical minimum and maximum values attainable, and visualised relative vulnerability based on those ranges. The theoretical minimum score was calculated as the lowest possible score attainable, and the theoretical maximum was calculated as the sum of the maximum value attained for each metric. The midpoint of each range (risk and threat) was used to divide the data into four quadrants: 1) High Risk-High Threat; 2) High Risk-Low Threat; 3) Low Risk-High Threat; 4) Low Risk-Low Threat, which correspond to four categories of erosion vulnerability. Beaches were then split into sites with and without *Casuarina* trees, and the vulnerability scores compared using a Kruskal Wallis Test (because assumptions of normality and heteroscedasticity were violated: $p < 0.05$ in both cases).

Table 1
Scoring categories for the risk and threat indicators that together comprise the beach vulnerability score.

Beach vulnerability			
Risk Indictors	Score	Threat Indicators	Score
1) Backshore width *Based on the width from the high tide line to the foot of the primary dune or the first line of coastal development*	**1** = > 26 m **2** = 18–26 m **3** = 12–18 m **4** = 9–12 m **5** = 0–9 m * *Training data set*	**1) Projected sea-level rise** *Average increase from < the 2020's to the 2090's for both the 4.5 and 8.5 RCP scenarios*	**1** = 0.33–0.36 m **2** = 0.36–0.38 m **3** = 0.38–0.40 m **4** = 0.40–0.48 m **5** = > 0.48 m * *Training data set*
2) Beach exposure *Based on surf zone entropy, geomorphological or other structures that may offer shelter and wave direction*	**0** = completely sheltered (blocked by a rocky reef or breakwater) **1** = sheltered **2** = semi-exposed, more sheltered than exposed **3** = semi-exposed, more exposed than sheltered **4** = fully exposed *Harris et al. (2011)	**2) Storm frequency** *Storm frequency refers to line density of the NOAA archived storm track data (1848 to 2018)* per 250 km^2 *(line density tool ArcMap 10.5.1)*	**1** = 0–0.031 **2** = 0.032–0.077 **3** = 0.078–0.126 **4** = 0.127–0.181 **5** = 0.182–0.247 **6** = 0.248–0.0326 **7** = 0.0327–0.412 **8** = 0.413–0.562 **Quantile range from line density raster*
3) Modal Beach energy *Based on the sum of the modal wave height, wind speed and tidal range*	**Waves** **0** = No waves **1** = Calm (Ripples, 0–0.1 m) **2** = Smooth (0.1–0.5 m) **3** = Slight (0.5–1.25 m) **4** = Moderate (1.25–2.5 m) **5** = Rough (2.5–4 m) **6** = Very rough (4–6 m) **7** = High (6–9 m) **8** = Very High (9–14 m) **9** = Phenomenal (> 14 m) **Douglas Sea Scale* **Wind** **0** = Calm (< 1 knots) **1** = Light air (1–3 knots) **2** = Light breeze (4–6 knots) **3** = Gentle breeze (7–10 knots) **4** = Moderate (11–16 knots) **5** = Fresh breeze (17–21 knots) **6** = Strong breeze (22–27) **7** = Moderate gale (28–33 knots) **8** = Fresh gale (34–40 knots) **9** = Strong gale (41–47 knots) **10** = Whole gale (48–55 knots) **11** = Storm (56–65 knots) **12** = Hurricane (> 65 knots) * *Beaufort scale* **Tide** 1 = micro-tidal (< 2 m) 2 = meso-tidal (2–4 m) 3 = macro-tidal (> 4 m) *Gornitz et al. (1994)	**3) Storm intensity** *Storm intensity refers to the maximum sustained winds (knots) extracted* per km^2 *from the NOAA archived storm track data (1848 to 2018)*	**1** = 0 - 25 knots **2** = 26–29 knots **3** = 30 - 34 knots **4** = 35 - 45 knots **5** = 46 - 55 knots **6** = 56- 70 knots **7** = 71 - 85 knots **8** = 86–185 knots **Quantile range from* max *wind speed raster*
4) State of the dunes system *Based on the condition of the dune system*	1 = multiple sand dune ridges 2 = single sand dune ridge 3 = developed or no dunes * Mclaughlin and Cooper (2010)	**4) Coastal development** *Based on location, intensity and extent.*	**Location of development** **0** = None **1** = Secondary dune **2** = Foredune **3** = Backshore **Intensity of development** **0** = None **1** = Low **2** = Moderate **3** = High **Extent of development** **0** = None **1** = 1/3 **2** = 2/3 **3** = 3/3 *Harris (2008)

2.3.4. Risk indicators

2.3.4.1. Backshore width. Google Earth (Pro 7.3.0.3832) was used to measure Backshore Width (BSW) in meters for the training data set and the study sites. BSW was defined as the section between the normal/modal high tide drift line and the edge of high-shore structures, be it primary dunes or development. Three measurements were taken for BSW, irrespective of beach shape and beach length: one in the middle, and two at either extreme end of the beach; these were averaged to provide a mean per beach. The training data set was used to define score categories, which were applied to the BSW of the study sites (Table 1; SM: Criteria and category development).

2.5.4.2. Beach exposure. Beach exposure was characterised using current and historical aerial imagery from Google Earth (Pro 7.3.0.3832). Four metrics were scored: surf-zone width (the larger the number of waves in the surf, the more exposed a beach is); image entropy (or image complexity; the greater the disrupted water flow, e.g., choppy waters, white foam canopies, presence of rip currents or side currents etc., the higher the energy of the surf-zone); protective structures (scoring the extent of rocky outcrops, reefs, or man-made structures, like harbours or groins, that shelter the beach from waves). The fourth indicator, wave direction/beach orientation, was scored from the average wave direction (perpendicular vs. parallel) across a week's predictions, randomly sub-sampled throughout the year (with eight predictions per day) on Windy (www.windy.com) to provide an estimate of exposure of the beach to waves. Wave direction is more stable than wind direction and a week's response was considered adequate following a verification test of known sites. Scores for beach exposure were obtained from Harris et al. (2011) and ranged 0–4, respectively indicating: a sheltered, calm beach; and a beach facing into oncoming waves with a wide surf zone comprising multiple breakers, no shoreward protection, and with substructure suggesting rip currents,

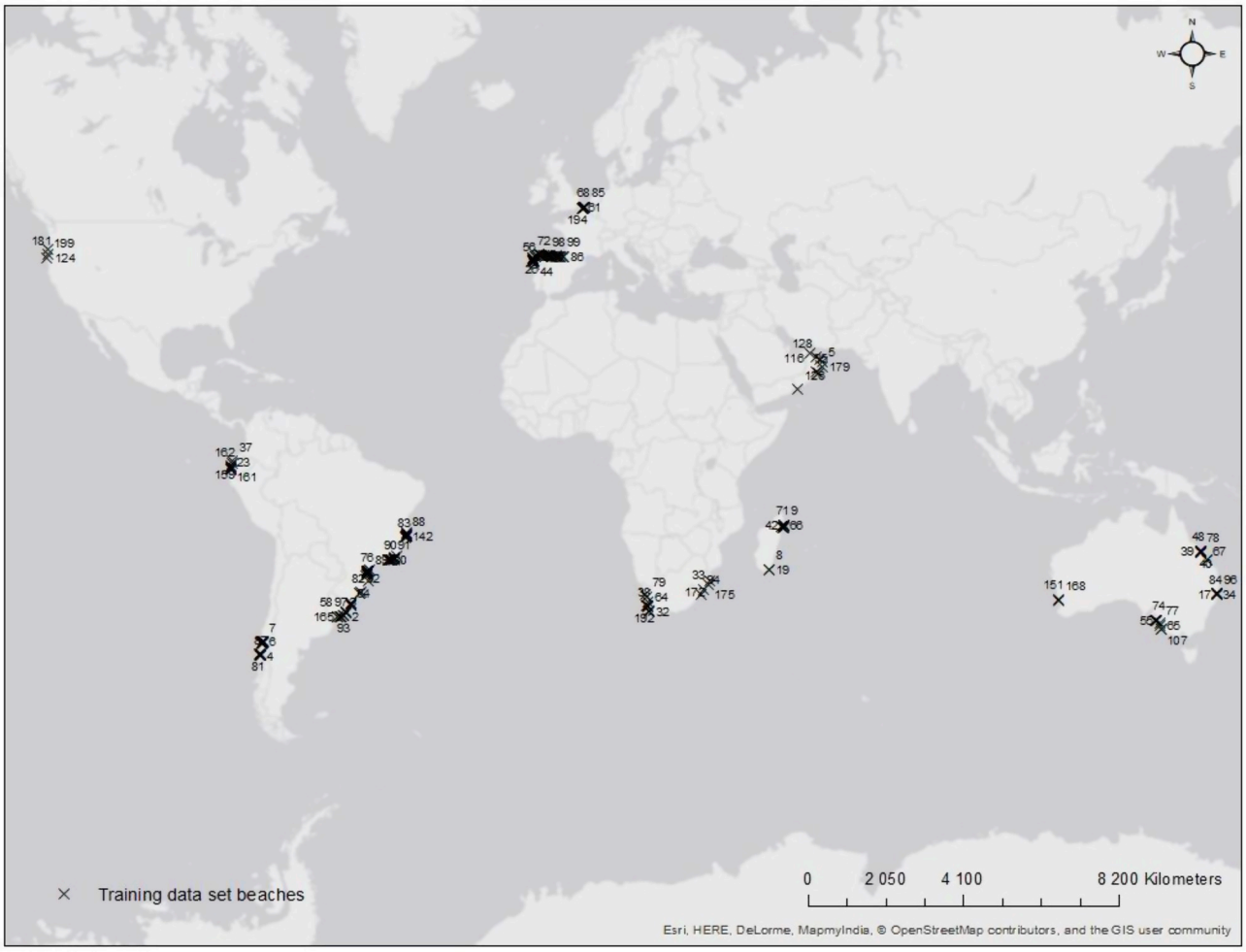
Training data set beaches
0 2 050 4 100 8 200 Kilometers
Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community

Fig. 2. Distribution of the beaches used in the training data set published by Defeo and McLachlan (2013) (n = > 200) to develop new risk/threat score categories.

reef or sand bars (Table 1). Half values were assigned if criteria were partially met. Where multiple beaches made up a single sea turtle rookery, each beach's orientation and protection was rated and the average exposure for the island or coastal area was calculated. I.e., for 22 beaches, which constituted 21 island beaches and the nesting beaches of Ras al Had, Oman, each beach was scored, and the mean calculated.

2.5.4.3. Modal beach energy. Modal beach energy was determined from the wave height, wind speed, and tide range scores. Data per beach were generated from the predicted wave height and wind speeds from Windy (www.windy.com) for 30 (haphazardly selected) days over a year, with eight recordings per day, (n = 240 per beach). The modal wind speed (knots) and wave height (m) value per beach were calculated. Tide range was obtained from a map of global tide ranges by Davies (1980), modified by Masselink and Hughes (2003).

Wave height, wind speed and tide range were first scored separately using existing published scales: Douglas Sea Scale ratings (Met Office, 2017) for waves, and Beaufort scale (Met Office, 2017; Royal Meteorological Society, 2017) for wind speed. Tide range was scored according to Gornitz et al. (1994), where micro-tidal environments (tide range < 2 m) were allocated a score of one (low risk), meso-tidal environments (tide range = 24 m) received a score of two (moderate risk) and macro-tidal environments (tide range > 4 m) received a score of three (high risk). To create a final score out of 24, the three categories of modal beach energy were summed, i.e. Modal beach energy = Wind speed score + Wave height score + Tide range score (Table 1).

2.5.4.4. State of the dune system. Google Earth (Pro 7.3.0.3832) was used to estimate the state of the dune system. In accordance with an existing ranking for dunes (Mclaughlin and Cooper, 2010), beaches with multiple sand dune ridges were scored as low risk (score = 1), beaches with a single sand dune ridge were scored moderate risk (score = 2) and beaches with no dunes or development in place of dunes were scored as high risk (score = 3).

2.3.5. Threat indicators

2.3.5.1. Coastal development. Coastal development was identified as any human alteration of the backshore, foredune or secondary dune. Any buildings or other hard structures (such as sea walls, loffelstein blocks, dolosse, rubble, harbour breakwaters, tourism amenities, roads, or rocks) that were visible on Google Earth Pro 7.3.0.3832 imagery were used to score the extent of coastal development following Harris (2008) (Table 1).

2.3.5.2. Sea-level rise. From the 47 general circulation models used in IPCC Fifth Assessment Report (2014) to evaluate scenarios of sea-level rise (SLR), the CSIRO ACESS1.0 model was selected. Outputs from this model provide consistent estimates across different regions, and over time (IPCC, 2014). The IPCC uses four Representative concentration pathways (RCPs) to drive models that provide projections of future climate states and their physical consequences. These include a stringent mitigation scenario (RCP2.6), two intermediate scenarios

(RCP4.5 and RCP6.0) and one extreme (but realistic) scenario with very high GHG emissions (RCP8.5), which portrays current global rate of GHG emission. Because RCP2.6 seems unattainable, we selected RCP4.5 as an optimistic mitigation scenario and RCP8.5 as business as-usual to evaluate likely future sea-level rise.

Two variables were extracted, "zos" and "zosga". "Zos" provided an estimate of sea level at any point in space and time relative to the ocean geoid (i.e., a value relative to mean global sea level). "Zosga" provided estimates of average global sea-level rise. Adding the two variables together provided an estimate of the sea level into the future for individual points of interest. Monthly estimates between 2006 and 2100 were extracted for each of the beaches in the training data set. We used bilinear interpolation among adjacent grid cells to obtain an estimate at the point of interest, rather than assuming the sea-level response is the same across the entire grid cell. The average SLR of the two scenarios (RCP4.5 and RCP6.0) from 2020 to 2100 was calculated per beach. Score categories were created using the training data set (See Fig. 2) and assigned to the study sites (Table 1; SM: Criteria and category development).

2.3.5.3. Storminess. Considering the inconsistencies of projected storm tracks at a coarse scale (global models) (IPCC, 2007), we used historical storm-track data (1848–2018) (Fig. 3) from the International Best Track Archive for Climate Stewardship (IBTrACS) Version 3 dataset from the NOAA National Climatic Data Center (data.noaa.gov), assuming similar storm conditions over the next 80 years. Because of the resolution in the data, both storm frequency and intensity scores were estimated.

2.3.5.3.1. Storm frequency. Storm frequency was calculated from the storm tracks of the last 50 years (1968–2018) in ArcMap 10.5.1. With a Cylindrical Equal Area projection, the line density tool was used to calculate a magnitude km^{-2} for the storm tracks that fell within a specified radius around each cell. The output raster cell size was 250 km^2 and the default radius from the line density tool was used. Storm track density was therefore calculated at a sub-regional level as a likely indication of storm frequency. The distribution of the storm track density (per km^2) values was plotted and split into 8 equal-interval groups to create score categories, which were assigned accordingly to the study sites. (Table 1; SM: Criteria and category development).

2.3.5.3.2. Storm intensity. Storm intensity values per 250 km^2 were obtained using 'feature to raster' tool in ArcMap 10.5.1, using the maximum sustained wind speed (knots) field from IBTrACS ver. 3 as the measure of storm intensity. From this, we generated a table of the storm intensity values for the study sites over the past 50 years (1968–2018). As for storm frequency, the distribution of the storm intensity values was plotted and split into 8 equal-interval groups to create score categories, which were assigned accordingly to the study sites. (Table 1; SM: Criteria and category development).

3. Results

3.1. Casuarina presence

Casuarina were present on 18 of the 50 turtle sites (Fig. 4); four of these beaches had native *Casuarina* trees, including a site in Brunei (BNX/#11), one in Indonesia (ID02/#21) and two in Australia (AU34/#11, AU70/#13). The 14 beaches with non-native/introduced *Casuarina* trees occurred along the Indian east coast, French Scattered Islands, north and west of Madagascar, on several Seychellois islands, and the South African north-east coast. One site in Sri Lanka, Rekawa (LK05/# 34), had a few introduced *Casuarina* covering less than 25% of the backshore. The seven sites in India, namely Devi river mouth (IN76/#32), Gahirmatha (Murali_01/#36), Kalingapatnam (IN102/#27), Mamallapuram (IN126/#28), Nagapattinam (IN137/#29), Rushikulya (Murali_02/#37), Srikakulam (IN200/#31) and Srikurmam (IN180/#30), had extensive *Casuarina* strips covering more than 25% of the backshore, very close to the high water mark. Non-native *Casuarina* occurrence was extensive on two of the Seychelles study sites: Cousin Island Special Reserve (SC08/#45) and Farquhar Island (SC52/#47). Photos and Google Earth imagery show *Casuarina* were present directly on the backshore for substantial portions of these two islands.

3.2. Vulnerability assessment

Half (n = 25 of 50 sites) of the backshores measured were less than 20 m wide, including a third (n = 17) that were very narrow (< 10 m). Beach exposure scores were spread across the range of the four constituent metrics of this indicator: many (46%) of the beaches had narrow surf zones and low image entropy (40%), indicating lower wave exposure, but some beaches (44%) had low wave protection. The study sites had low to moderate modal beach energy, with modal wind speed ranging between 2 and 23 knots, modal wave height ranging between 0.1 and 2.1 m and most of the beaches (n = 33) had a meso-tidal range (2–3 m). However, beaches often received a higher risk score because half the beaches had no dunes (n = 23). To view risk indicator data per beach, refer to SM, Table A2.1.

Overall, the beaches had comparatively higher vulnerability on account of physical beach features rather than from threats (Fig. 5; SM, Table A2.2). Half the beaches (n = 26) had low levels of development and were shown to be at low to moderate threat from storm frequency, where storm track density ranged from 0 to 0.14 km^{-2} and only two

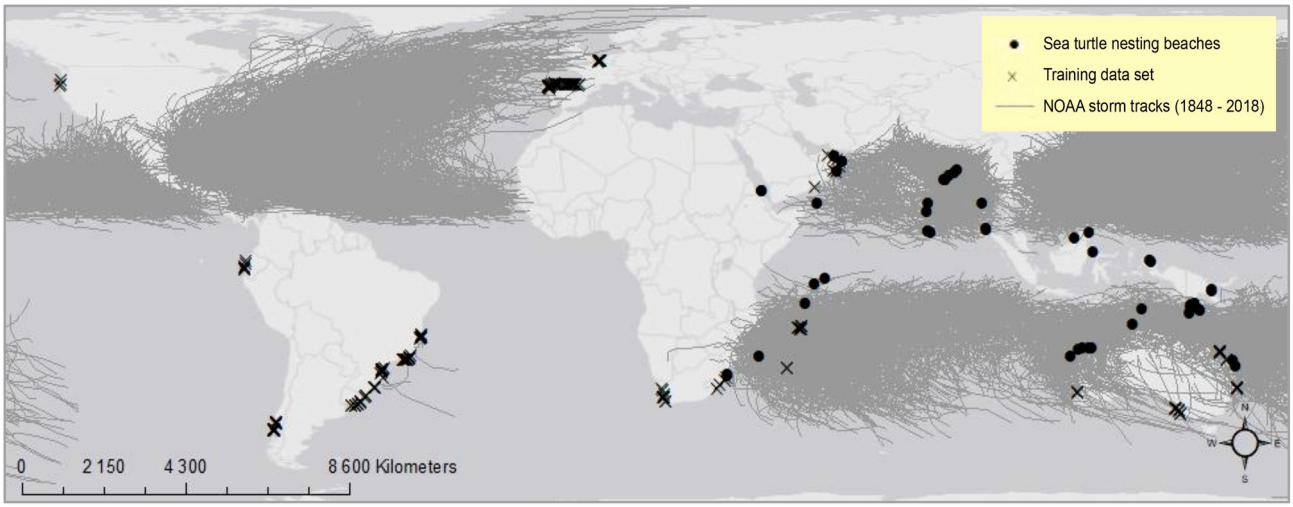

Fig. 3. Global distribution of storm tracks (1848–2018). Data from: IBTrACS Version 3 from the NOAA National Climatic Data Center.

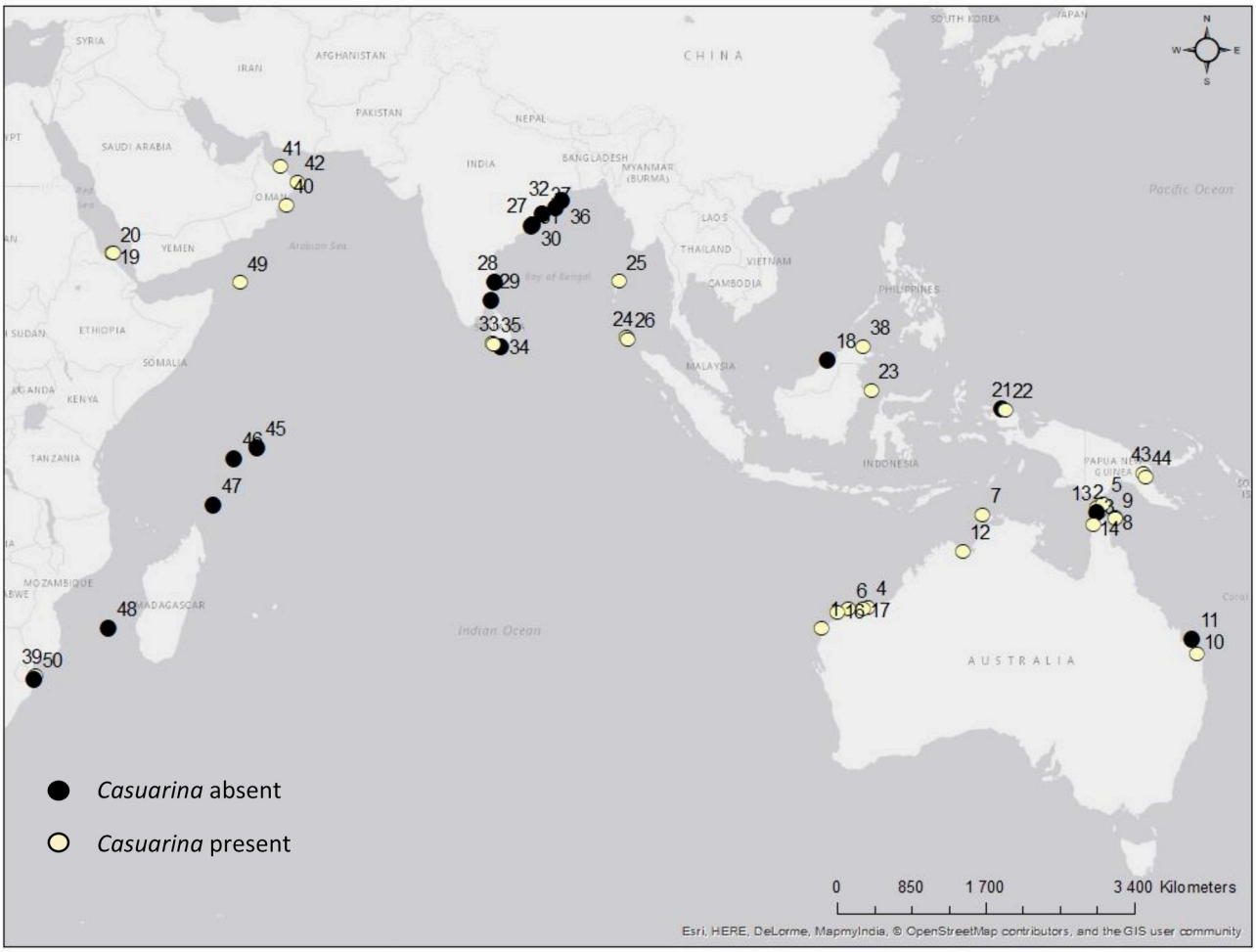

Fig. 4. Map depicting Casuarina presence for the study sites/nesting beaches.

Omani beaches were shown to be highly threatened by storm intensity: Dalmaniyat (OM11/#41) and Ras al Had (OM12/#42). For these beaches, the maximum sustained wind during recorded storm events was 77 knots, but most beaches were shown to be at low to moderate threat for storm intensity (maximum sustained wind range was 0–55 knots). All beaches (n = 50) predicted low to moderate sea-level rise, which ranged from 0.31 to 0.38 m by the year 2100.

To generate regionally appropriate vulnerability categories, the theoretical maximum of the x- and y-axis was set (threat = 2.86; risk = 4) (Fig. 5). This approach allowed for regional comparison among beaches, where beaches were spread across the four vulnerability categories; High Risk-High Threat, Low Risk-High Threat, High Risk-Low Threat and Low Risk-Low Threat. Based on the normalised score of paired risk (backshorewidth, beach exposure, modal beach energy, and dune state) and threat (coastal development, storm frequency, storm intensity and sea-level rise) indicators, two beaches were categorised as High Risk-High Threat: Cemetery beach (AU119/#4) and Bentota beach (LK02/#33). One beach, Barrow island (AU81/#16), was categorised as Low Risk-High Threat. However, half the beaches were categorised as Low Risk-Low Threat (n = 25 beaches) and the rest (n = 22 beaches) as High Risk-Low Threat. Overall, there was no significant difference in erosion vulnerability on beaches with *Casuarina* trees compared to those without (Kruskal-Wallis Test: $X^2_{44} = 42.62$, p = 0.5308).

The top 10 most vulnerable beaches based on the final vulnerability score (sum of the normalised risk and threat scores) included Cemetery beach (AU119/#4), Mamallapuram-Pondi beach (IN126/#28), Bentota beach (LK02/#33), Cousin Island Special Reserve (SC08/#45), Nagapattinam beach (IN137/#29), Wreck Island (AU34/#11), Brunei beach (BNX/#18), D'Arros Island and St. Joseph Atoll (SC10/#46), Kamiali Wildlife Management Area (PG04/#44) and Kosgoda beach (LK21/335) (Table 2). Half of these top ten most vulnerable beaches hosted *Lepidochelys olivacea* nesting females. Furthermore, non-native *Casuarina* trees were also most dominant on the rookery beaches for this species; overall 82% of the *L. olivacea* sites had non-native *Casuarina* present. The L. *olivacea* rookery with the highest number of nesting females (150 000–200 000) (Shanker et al., 2004), Devi river mouth, Orissa (IN76/#32), had exotic *Casuarina* trees present to an extent of more than 25% along the backshore. The two Sri Lankan beaches, Bentota (LK02/#33) and Kosgoda (LK21/#35) host multiple sea turtle nesting species, including *Lepidochelys olivacea* and *Chelonia mydas* (Ekanayake et al., 2010), while Bentota beach also hosts *Dermochelys coriacea* (Ekanayake et al., 2010) and Kosgoda beach has *Caretta caretta* (The Turtle Conservation Project: http://www.tcpsrilanka.org/download/Map.pdf). Non-native *Casuarina* trees were absent from both beaches.

Two of the top 10 most vulnerable beaches hosted *Eretmochelys imbricata* nesting females: the Seychellois islands, D'Arros Island and St. Joseph Atoll (SC10/#46), with the highest number of *E. imbricata* nesting females of the sites selected for this species (375 females annually) (Mortimer, 2005) and Cousin Island Special reserve (SC08/#45) with approximately 90 tagged females in 2008 (Nature, 2008). Both these islands had extensive *Casuarina* presence (> 25%) along the backshore. One *Caretta caretta* nesting beach, Wreck Island (AU34/#11), was included in the top 10 most vulnerable beaches with native *Casuarina* occurrence. One *Natator depressus* nesting site, Cemetery

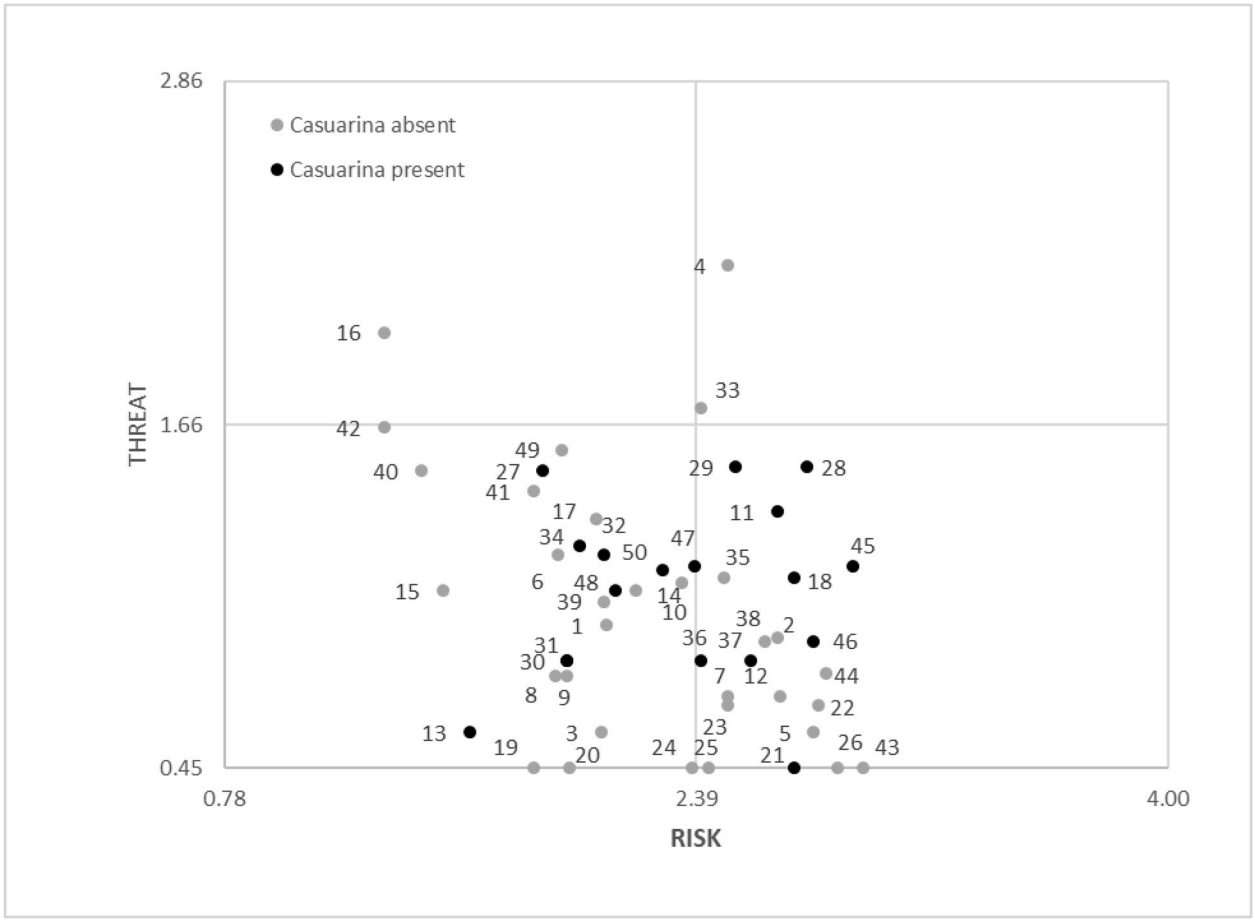

Fig. 5. Regional view of vulnerability categories for study sites where the theoretical maximum (threat = 2.86; risk = 4) was set to generate regionally appropriate categories. Vulnerability categories for study sites based on paired risk (backshore width, beach exposure, modal beach energy and dune state) and threat (coastal development, storm frequency, storm intensity and sea-level rise) scores. The four categories are: High Risk-High Threat, High Risk-Low Threat, Low Risk-High Threat and Low Risk-Low Threat. Kruskal-Wallis of CVI scores between Casuarina present vs absent Kruskal-Wallis X244 = 42.62, p = 0.5308.

beach (AU119/#4), identified as the most vulnerable beach in the study, with approximately 400 nesting females annually (Prince, 2009), had no *Casuarina* trees present.

4. Discussion

The aim of this study was to estimate erosion vulnerability as a composite measure of risk (physical beach characteristics) and threat (natural and anthropogenic stressors) for important sea turtle beaches in the IOSEA region, and to determine whether erosion vulnerability was higher on beaches backed by non-native *Casuarina* trees. The premise is that, if the latter is true, stabilising backshores with *Casuarina* ultimately contributes to enhancing coastal vulnerability rather than protecting shorelines with little ecological impact – as is currently intended and assumed. We found that planting non-native *Casuarina* is a fairly common management response in the IOSEA region, with these trees present on 28% of the sea turtle rookeries selected for this study. Several of these beaches showed high vulnerability to erosion because of physical beach characteristics, largely backshore condition and development. Of the top 10 most vulnerable beaches, half have introduced *Casuarina* on the backshore. However, contrary to our prediction, beaches with *Casuarina* present did not have significantly higher vulnerability to erosion compared to beaches without *Casuarina*; other indicators seem to be stronger drivers of erosion vulnerability.

4.1. Erosion vulnerability of IOSEA turtle nesting beaches

A regional view of the study sites showed comparatively high vulnerability for two beaches: Cemetery beach (#4) and Bentota (#33) beach. Both these beaches were distributed within the High Risk-High Threat category. However, since the nests at Bentota beach are relocated to minimise threat of poaching/predation and also for purposes for ecotourism, the eggs are protected (Ekanayake et al., 2010). One beach, Barrow Island (#16) was categorised as Low Risk-High Threat, with 22 beaches categorised as High Risk-Low Threat and 25 beaches within the Low Risk-Low Threat category. Because the sea turtle nesting beaches showed low to moderate modal beach energy, sea-level rise, and storm frequency and intensity, the High Risk-High Threat beaches generally had the following characteristics: narrow backshore width (< 10 m), high beach exposure, no dune system and/or high(er) levels of coastal development. These characteristics increase the erosion potential of beaches. Backshore or dry beach width acts a buffer zone against wave action, therefore wider backshore sections will offer more protection (Anfuso and Martínez Del Pozo, 2009, RANGEL-Buitrago and Anfuso, 2015), indicating the narrow backshore sections would make the majority of the sea turtle nesting beaches more vulnerable to oncoming waves (Rizzo et al., 2017). High beach exposure means beach orientation and low/no physical protection allows for direct wave action making beaches more susceptible to sand loss (Bryan et al., 2001; Mclaughlin and Cooper, 2010; Goodhue et al., 2012). If the dune system is compromised through development or if no dunes occur, sand loss might be permanent as dunes act as sand reserves that replenish the

Table 2
Vulnerability score (normalised risk score + normalised threat score) assigned to each of the study sites. Sites are listed in order of vulnerability (highest to lowest). The top 10 most vulnerable beaches are shaded in light grey.

Site number	Country	Site ID	*Casuarina* present /absent	Beaches	Species	Summed sk score	Summed	Vulnerability
4	Australia	AU119	Absent	Cemetery Beach	*Natator depressus*	2.50	2.21	4.71
28	India	IN126	Present	Mamallapuram - Pondi	*Lepidochelys olivacea*	2.77	1.51	4.27
33	Sri Lanka	LK02	Absent	Bentota	*Chelonia mydas* *Lepidochelys olivacea* *Dermochelys coriacea*	2.41	1.71	4.12
45	Seychelles	SC08	Present	Cousin Island Special Reserve	*Eretmochelys imbricata*	2.93	1.16	4.08
29	India	IN137	Present	Nagapattinam	*Lepidochelys olivacea*	2.53	1.51	4.03
11	Australia	AU34	Present	Wreck Island	*Caretta caretta*	2.67	1.35	4.02
18	Brunei Darussalam	BNX	Present	Brunei	*Lepidochelys olivacea*	2.73	1.12	3.84
46	Seychelles	SC10	Present	D'Arros Island and St. Joseph Atoll	*Eretmochelys imbricata* *Chelonia mydas*	2.79	0.89	3.69
44	Papua New Guinea	PG04	Absent	Kamiali Wildlife Management Area	*Dermochelys coriacea*	2.83	0.78	3.62
35	Sri Lanka	LK21	Absent	Kosgoda	*Chelonia mydas* *Caretta caretta* *Lepidochelys olivacea*	2.48	1.12	3.60
2	Australia	AU10	Absent	Dayman Island	*Eretmochelys imbricata*	2.67	0.91	3.58
47	Seychelles	SC52	Present	Farquhar Group	*Chelonia mydas* *Eretmochelys imbricata*	2.38	1.16	3.54
38	Malaysia	MY02	Absent	Turtle Islands, Sabah	*Chelonia mydas* *Eretmochelys imbricata*	2.63	0.89	3.52
49	Yemen	YE01	Absent	Abalhan Protected Area/ Socotra Man and Biosphere Reserve	*Caretta caretta*	1.93	1.57	3.50
22	Indonesia	ID14	Absent	Warmon	*Dermochelys coriacea*	2.81	0.67	3.48
10	Australia	AU33	Absent	Woongarra coast including Mon Repos	*Caretta caretta*	2.34	1.10	3.44
50	South Africa	ZA01	Present	Mabibi to Kosi Lake/Bhanga Nek	*Dermochelys coriacea* *Caretta caretta*	2.28	1.14	3.42
43	Papua New Guinea	PG02	Absent	Busama (Buli)	*Dermochelys coriacea*	2.96	0.45	3.41
37	India	Murali_02	Present	Rushikulya	*Lepidochelys olivacea*	2.58	0.83	3.40
12	Australia	AU39	Absent	Cape Domett	*Natator depressus*	2.68	0.70	3.38
17	Australia	AU82	Absent	Mundabullangana Beach	*Natator depressus*	2.05	1.33	3.38
5	Australia	AU15	Absent	Long Island	*Eretmochelys imbricata*	2.79	0.58	3.37
27	India	IN102	Present	Kalingapatnam/Vamsadhara	*Lepidochelys olivacea*	1.87	1.49	3.36
26	India	IN05	Absent	Galathea Beach, Great Nicobar Island	*Dermochelys coriacea*	2.88	0.45	3.33
16	Australia	AU81	Absent	Barrow Island	*Natator depressus* *Chelonia mydas* *Eretmochelys imbricata*	1.33	1.98	3.30
32	India	IN76	Present	Devi River mouth, Orissa	*Lepidochelys olivacea*	2.08	1.20	3.28
14	Australia	AU71	Absent	Flinders Beach	*Natator depressus*	2.18	1.08	3.26
41	Oman	OM11	Absent	Dalmaniyat	*Eretmochelys imbricata*	1.83	1.42	3.26
36	India	Murali_01	Present	Gahirmatha	*Lepidochelys olivacea*	2.41	0.83	3.23
34	Sri Lanka	LK05	Present	Rekawa	*Chelonia Myda* *Caretta caretta* *Dermochelys coriacea* *Lepidochelys olivacea* *Eretmochelys imbricata*	1.99	1.23	3.22
7	Australia	AU284	Absent	Cape Van Diemen	*Lepidochelys olivacea*	2.50	0.70	3.20
48	French Southern Territories	TF03	Present	Europa	*Chelonia mydas*	2.11	1.08	3.19
21	Indonesia	ID02	Present	Jamursba Medi	*Dermochelys coriacea*	2.73	0.45	3.18
23	Indonesia	ID27	Absent	Bilang-Bilangan	*Chelonia mydas*	2.50	0.67	3.17
6	Australia	AU27	Absent	Rosemary Island	*Eretmochelys imbricata*	1.92	1.20	3.12
39	Mozambique	MZ07	Absent	Malongane	*Caretta caretta*	2.08	1.03	3.11
1	Australia	AU08	Absent	Bungelup	*Caretta caretta*	2.08	0.95	3.03

(*continued on next page*)

Table 2 (*continued*)

Site number	Country	Site ID	*Casuarina* present /absent	Beaches	Species	Summed sk score	Summed	Vulnerability
42	Oman	OM12	Absent	Ras al Had	*Chelonia mydas*	1.33	1.64	2.97
40	Oman	OM02	Absent	Masirah	*Caretta caretta*	1.45	1.49	2.94
25	India	IN04	Absent	Cuthbert Bay	*Lepidochelys olivacea*	2.43	0.45	2.88
24	India	IN02	Absent	Beaches straddling the Alexandria and Dagmar Rivers	*Dermochelys coriacea*	2.38	0.45	2.83
30	India	IN180	Present	Srikurmam	*Lepidochelys olivacea*	1.95	0.83	2.78
31	India	IN200	Present	Srikakulam	*Lepidochelys olivacea*	1.95	0.83	2.78
9	Australia	AU309	Absent	Raine Island, north Great Barrier Reef	*Chelonia mydas*	1.95	0.78	2.73
8	Australia	AU300	Absent	Moulter Cay, north Great Barrier Reef	*Chelonia mydas*	1.91	0.78	2.68
3	Australia	AU11	Absent	Hawkesbury (Warral) Island	*Eretmochelys imbricata*	2.07	0.58	2.64
15	Australia	AU80	Absent	Wild Duck	*Natator depressus*	1.53	1.07	2.60
20	Eritrea	ER02	Absent	Mojeidi Island	*Eretmochelys imbricata*	1.96	0.45	2.41
19	Eritrea	ER01	Absent	Aucan Island	*Eretmochelys imbricata*	1.83	0.45	2.28
13	Australia	AU70	Present	Crab Island	*Natator depressus*	1.62	0.58	2.19

beach's sand budget (Tinley, 1985; Tsoar, 2001, Abuodha, 2006).

Although *Casuarina* presence may contribute to erosion vulnerability in some cases, the beaches in this study seem to be more vulnerable to erosion a result of physical characteristics that increase the risk of beach inundation/flooding processes (Gornitz et al., 1994). Certainly, there is a broader distribution of scores along the risk axis than threat axis, suggesting that if the threat regime had to increase, many beaches would become substantially more vulnerable to erosion. Of the threat factors, coastal development is the only one that can be controlled by beach managers. Management strategies should therefore consider options that reduces the risk of erosion and episodic inundation, such as restoring degraded dunes and/or implementing conservative set-back lines that reduce the risk of damage to human settlements and promotes conservation of the sandy beach ecosystem.

4.2. Implications of Casuarina presence for nesting turtles

Even though *Casuarina* trees may not strongly impact nesting turtles through an erosion-vulnerability mechanism, there are other effects of these trees that need to be considered prior to planting bio-shields behind rookeries. Sea turtles use the backshore to nest close to, in or under vegetation (Hays et al., 1995). Vegetation thus has a direct effect on the incubation environment and ultimately, hatchling sex ratios and hatchling success (Wood and Bjorndal, 2000). The effect of *Casuarina* trees on soil characteristics, such as pH (Batish et al., 2001) and temperature (Chaudari et al., 2009), may significantly change parameters required for successful egg incubation or hatching. Furthermore, the significantly lower temperature underneath *Casuarina* trees may have implications for sex ratios of sea turtle hatchlings (because sea turtles have temperature-dependent sex determination), with more males being produced in the presence of *Casuarina* trees (Chaudari et al., 2009). Furthermore, a toppled *Casuarina* tree can provide a physical barrier to female turtles as they haul up the beach to nest; and the root system can provide a barrier to females as they attempt to dig their nests. For example, Chaudari et al. (2009) highlighted the potential negative impacts of *Casuarina* trees on *L. olivacea* turtles along the Tamil Nadu coast, showing that fewer turtles nest in the presence of *Casuarina* trees.

Of the top 10 most vulnerable beaches, half had non-native *Casuarina* trees present. Mamallapuram – Pondi beach (#126) and Nagapattinam (#29) are two high nesting density *L. olivacea* and with non-native *Casuarina* trees comprising more than 25% along the backshore or dunes. Furthermore, non-native *Casuarina* occurrence is most common on *L. olivacea* study sites of the Western Indian Ocean (attributed to large scale plantations of *C. equisetifolia* on the Indian coastline), including Devi river mouth, where 150 000–200 000 female turtles nest (Shanker et al., 2004). This is of concern because *L. olivacea* populations of the west and north-east Indian Ocean are some of the least resilient/most vulnerable marine turtle Regional Management Units (RMUs). These L. *olivacea* populations were categorised as vulnerable to climate change because of rookery vulnerability (the likelihood of functional rookeries becoming extirpated) and non climate-related threats, such as fisheries, take, coastal development and pollution/pathogens (Fuentes et al., 2013). Staged *Casuarina* removal, at least as far inland as turtles' nest, should thus be carefully considered by managers, especially on beaches where *L. olivacea* nest.

4.3. CVI method: a way forward

The CVI method involves scoring a suite of indicators and can provide a good regional overview of general patterns. Ideally, though, this should be followed with quantitative, local-scale assessments (ideally with data from field surveys) to establish if *Casuarina* trees are contributing to erosion at smaller scales, serving also to ground-truth the results. In this case, there are several indicators that were omitted in our study due to data limitations at an ocean-basin scale, but that could

be very informative in more detailed, local assessments. These include elevation (Kumar et al., 2010; Gornitz et al., 1994) and shoreline change (Thieler and Hammer-Klose, 2000; Boruff et al., 2005; Pendleton et al., 2010). We recommend including long-term data on these indicators because shoreline change gives a precise indication of whether a beach has been eroding or accreting (Corbella and Stretch, 2012) and elevation gives a good indication of a beach's ability to withstand or recover from episodic events (Abuodha, 2006). Future studies should also consider indicators that address the recovery potential of beaches. For example, Pethick and Crooks (2000) related disturbance-event frequency to relaxation time (the time taken for the littoral component to recover its shape), thereby providing an approximation of temporal variability of coastal features.

4.4. Conclusion

Research to date on the sustainability of using exotic *Casuarina* trees in coastal regions has demonstrated physical impacts on sandy beaches (Morton, 1980; Jadhav and Gaynar, 1995; Gordon, 1998; Batish et al., 2001; Patil et al., 2002; Sealey, 2006; Chaudari et al., 2009; Wheeler et al., 2011; Hardman et al., 2012). Non-native *Casuarina* were present on beaches in the IOSEA region, and planting these trees seems to be a popular management action. Although the regional CVI analysis did not show more vulnerability to erosion on beaches backed by *Casuarina*, it is important to recognise that these trees may pose other threats to nesting sea turtles. These threats include changing the nest-incubation environment, and serving as barriers to female turtles during nesting, especially if the trees have toppled over. Furthermore, the CVI was applied at an ocean-basin scale, and local-scale indicators and data may provide more refined insights on the effect of these trees on beaches. Therefore, managers and other decision-makers need to carefully consider the impacts of non-native *Casuarina* bio-shields before planting these trees, especially if the beaches are turtle rookeries. In the IOSEA region, *L. olivacea* rookeries are especially vulnerable to the impacts of *Casuarina*, and mitigation measures, such as staged removal of these trees, might be a prudent course of action. Maintaining and restoring intact dune cordons behind beaches has multiple benefits (e.g., protection for people and infrastructure, and conservation of ecological patterns and processes) and thus ultimately, implementing conservative setback lines is the most ideal approach to managing sandy shores.

Acknowledgements

Funding for this project was provided by a Nelson Mandela University Science Faculty Grant. The National Research Foundation and Nelson Mandela University provided financial support to DDV. Opinions expressed, and conclusions arrived at, are those of the authors and are not necessarily to be attributed to the National Research Foundation.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2019.03.015.

References

Abuodha, P.A., Woodroffe, C.D., 2006. Assessing vulnerability of coasts to climate change: a review of approaches and their application to the Australian coast. In: GIS for the Coastal Zone: A selection of Papers from CoastGIS 2006:458.
Anfuso, G., Martínez Del Pozo, J.Á., 2009. Assessment of coastal vulnerability through the use of GIS tools in south Sicily (Italy). Environ. Manag. 43, 533–545.
Batish, D.R., Singh, H.P., Kohli, R.K., 2001. Vegetation exclusion under *Casuarina equisetifolia* L.: does allelopathy play a role? Commun. Ecol. 2, 93–100.
Benassai, G., DI Paola, G., Aucelli, P.P.C., 2015. Coastal risk assessment of a micro-tidal littoral plain in response to sea level rise. Ocean Coast Manag. 104, 22–35.
Bhalla, R.S., 2007. Do bio-shields affect tsunami inundation? Curr. Sci. 93, 831–833.
Boruff, B.J., Emrich, C., Cutter, S.L., 2005. Erosion hazard vulnerability of US coastal counties. J. Coast. Res. 215, 932–942.
Brown, A.C., Mclachlan, A., 2002. Sandy shore ecosystems and the threats facing them: some predictions for the year 2025. Environ. Conserv. 29, 62–77.
Bryan, B.A., Harvey, N., Belperio, T., Bourman, B., 2001. Distributed process modelling for regional assessment of coastal vulnerability to sea-level rise. Environ. Model. Assess. 6, 57–65.
Buehler, C., Rodgers, J., 2012. Soil property differences between invaded casuarina (*Casuarina equisetifolia* L.) sites and non-casuarina sites in the Bahamas. Phys. Geogr. 33, 574–588.
Bulleri, F., Chapman, M.G., 2010. The introduction of coastal infrastructure as a driver of change in marine environments. J. Appl. Ecol. 47, 26–35.
Canadell, J.G., Pataki, D.E., Pitelka, L.F. (Eds.), 2007. Terrestrial Ecosystems in a Changing World. Springer Science & Business Media.
CAB International, 2000. Forestry Compendium Global Module. CAB International, Wallingford, UK.
Charlier, R.H., Chaineux, M.C.P., Morcos, S., 2005. Panorama of the history of coastal protection. J. Coast. Res. 21, 79–111.
Chaudari, S., Prasad, K.V., Shanker, K., 2009. Impact of *Casuarina* Plantations on Olive Ridley Turtle Nesting along the Northern Tamil Nadu Coast , India. ATREE, Bangalore and MCBT, Mamallapuram, India, pp. 1–53.
Corbella, S., Stretch, D.D., 2012. Decadal trends in beach morphology on the east coast of South Africa and likely causative factors. Nat. Hazards Earth Syst. Sci. 12, 2515–2527.
Crowell, M., Edelman, S., Coulton, K., Mcafee, S., 2007. How many people live in coastal areas? J. Coast. Res. 23 (iii–vi).
Danielsen, F., Sørensen, M.K., Olwig, M.F., Selvam, V., Parish, F., Burgess, N.D., Hiraishi, T., Karunagaran, V.M., Rasmussen, M.S., Hansen, L.B., Quarto, A., 2005. The Asian tsunami: a protective role for coastal vegetation. Science 310, 643.
Das, S., Sandhu, H., 2014. Role of exotic vegetation in coastal protection. Econ. Pol. Wkly. 49, 43.
Davies, J.L., 1980. Geographical Variation to Coastal Development, second ed. Longman (Pearson Education Limited.
De Zoysa, M., 2008. *Casuarina* coastal forest shelterbelts in Hambantota City, Sri Lanka: assessment of impacts. Small Scale For. 7, 17–27.
Defeo, O., Mclachlan, A., 2013. Global patterns in sandy beach macrofauna: species richness, abundance, biomass and body size. Geomorphology 199, 106–114.
Defeo, O., Mclachlan, A., Schoeman, D.S., Schlacher, T.A., Dugan, J., Jones, A., Lastra, M., Scapini, F., 2009. Threats to sandy beach ecosystems: a review. *Estuarine*. Coast. Shelf Sci. 81, 1–12.
Ekanayake, E.M.I., Rajakaruna, R.S., Kapurusinghe, T., Saman, M.M., Rathnakumara, D.S., Samaraweera, P., Ranawana, K.B., 2010. Nesting behaviour of the green turtle at Kosgoda rookery, Sri Lanka. Ceylon J. Sci. Biol. Sci. 39, 109–120.
Fuentes, M.M.P.B., Pike, D.A., Dimatteo, A., Wallace, B.P., 2013. Resilience of marine turtle regional management units to climate change. Glob. Chang. Biol. 19, 1399–1406.
Glick, P., Stein, B.A., Edelson, N.A. (Eds.), 2011. Scanning the Conservation Horizon: A Guide to Climate Change Vulnerability Assessment. National Wildlife Federation, Washington, D.C.
Goodhue, N., Rouse, H., Ramsay, D., Bell, R., Hume, T., Hicks, M., 2012. Coastal Adaptation to Climate Change: Mapping a New Zealand Coastal Sensitivity Index. A Report Prepared as Part of the Coastal Adaptation to Climate Change Project under Contract (CO1XO802) to MBIE. NIWA, Hamilton, New Zealand, pp. 43p.
Gordon, D.R., 1998. Effects of invasive, non-indigenous plant species on ecosystem processes: lessons from Florida. Ecol. Appl. 8, 975–989.
Gornitz, V.M., Daniels, R.C., White, T.W., Birdwell, K.R., 1994. The development of a coastal risk assessment database: vulnerability to sea-level rise in the U.S. southeast. J. Coast. Res. Special issue 12, 327–338.
Halpern, B.S., Frazier, M., Potapenko, J., Casey, K.S., Koenig, K., Longo, C., Lowndes, J.S., Rockwood, R.C., Selig, E.R., Selkoe, K.A., Walbridge, S., 2015. Spatial and temporal changes in cumulative human impacts on the world's ocean. Nat. Commun. 6, 7615.
Hardman, C.J., Williams, S., Manco, B.N., Hamilton, M.A., 2012. Predicting the potential threat of *Casuarina equisetifolia* to three endemic plant species on the Turks and Caicos Islands. Oryx 46, 204–212.
Harris, L.R., 2008. The Ecological Implications of Sea-Level Rise and Storms for Sandy Beaches in Kwazulu-Natal. MSc thesis. Nelson Mandel Metropolitan University, Port Elizabeth, South Africa.
Harris, L.R., Nel, R., Schoeman, D.S., 2011. Mapping beach morphodynamics remotely: a novel application tested on South African sandy shores. Estuar. Coast Shelf Sci. 92, 78–89.
Hays, G.C., Mackay, A., Adams, C.R., Mortimer, J.A., Speakman, J.R., Boerema, M., 1995. Nest site selection by sea turtles. J. Mar. Biol. Assoc. U. K. 75, 667–674.
IPCC, 2007. Climate change 2007: The physical science basis. In: Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, pp. 996.
IPCC, 2014. Climate change 2014: Synthesis Report. In: Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, pp. 1–167.
Jadhav, B.B., Gaynar, D.G., 1995. Effect of *Casuarina equisetiolia* J.R. leaf litter leachates on germination and seedling growth of rice and cowpea. Allelopathy J. 2, 105–108.
Klukas, R.W., 1969. Exotic Terrestrial Plants in South Florida with Emphasis on Australian Pine (*Casuarina equisetifolia*). Everglades National Park, Homestead, Florida Report, (33030).
Kumar, T., Mahendra, R., Nayak, S., Radhakrishnan, K., Sahu, K., 2010. Coastal vulnerability assessment for Orissa State, east coast of India. J. Coast. Res. 26, 523–534.
Li, K., Li, G.S., 2011. Vulnerability assessment of storm surges in the coastal area of

Guangdong Province. Nat. Hazards Earth Syst. Sci. 11, 2003–2010.
Masselink, G., Hughes, M.G., 2003. Introduction to Coastal Processes and Geomorphology. Oxford University Press.
Mathiventhan, T., Jayasingam, T., 2014. Coastal green belt in Batticaloa district, Sri Lanka: is *Casuarina* a success? Int. J. Mar. Sci. 4. https://doi.org/10.5376/ijms.2014.04.0055.
Mazzotti, F.J., Ostrenko, W., Smith, A.T., 1981. Effects of the exotic plants *Melaleuca quinquenervia* and *Casuarina equisetifolia* on small mammal populations in the eastern Florida Everglades. Fla. Sci. 44, 65–71.
Mclaughlin, S., Cooper, J.A.G., 2010. A multi-scale coastal vulnerability index: a tool for coastal managers? Environ. Hazards 9, 233–248.
Mclaughlin, S., Mckenna, J., Cooper, J.A.G., 2002. Socio-economic data in coastal vulnerability indices: constrains and opportunities. J. Coast. Res. 497, 487–497.
Met Office, 2017. Beaufort wind force scale. [Online] Available at. https://www.metoffice.gov.uk/binaries/content/assets/mohippo/pdf/b/7/fact_sheet_no._6.pdf, Accessed date: 23 October 2017.
Mortimer, J.A., 2005. Sea turtles of D'arros island and St. Joseph Atoll: status and recommendations. In: Engelhardt, U. (Ed.), Proceedings of a Scientific Symposium Held at the D'Arros Research Centre 15–17 April 2005. Seychelles: D'Arros Research Centre, Technical Report No. 1.
Morton, J.F., 1980. The Australian pine or beefwood (*Casuarina equisetifolia* L.), an invasive 'weed' tree in Florida. Proc. Fla. State Hortic. Soc. 93, 87–95.
Mukherjee, N., Balakrishnan, M., Shanker, K., 2009. Bioshields and ecological restoration in tsunami-affected areas in India. In: Moksness, E., Dahl, E., Støttrup, J. (Eds.), Integrated Coastal Zone Management. John Wiley and Sons, Oxford, pp. 131–144.
Nature, seychelles, 2008. Hawksbill nesting on Cousin island, Seychelles. SWOT Report—The State of the World's Sea Turtles, vol 3 (2008).
Patil, R.H., Hunshal, C.S., Itnal, C.J., 2002. Effect of casuarina litter leachates on crops. Allelopathy J. 10, 141–146.
Pendleton, E.A., Thieler, E.R., Williams, S.J., 2010. Importance of coastal change variables in determining vulnerability to sea- and lake-level change. J. Coast. Res. 261, 176–183.
Pethick, J., Crooks, S., 2000. Development of a coastal vulnerability index: a geomorphological perspective. Environ. Conserv. 27, 359–367.
Pethick, J., 2001. Coastal management and sea-level rise. Catena 42, 307–322.
Potgieter, L.J., Richardson, D.M., Wilson, J.R.U., 2014a. *Casuarina*: biogeography and ecology of an important tree genus in a changing world. Biol. Invasions 16, 609–633.
Potgieter, L.J., Richardson, D.M., Wilson, J.R.U., 2014b. *Casuarina cunninghamiana* in the Western Cape, South Africa: determinants of naturalisation and invasion, and options for management. South Afr. J. Bot. 92, 134–146.
Western Australia Marine Turtle ProgramPrince, R.I.T., 2009. Flatback nesting in western Australia: personal communication. SWOT Report—The State of the World's Sea Turtles, *v*ol 4 (2009).
RANGEL-Buitrago, N., Anfuso, G., 2015. Risk assessment of storms in coastal zones: case studies from Cartagena (Colombia) and Cadiz (Spain). In: SpringerBriefs in Earth Sciences Book Series. Springer, Berlin, pp. pp7–14.
Rizzo, A., Aucelli, C.P.C., Gracia, F.J., Anfuso, G., 2017. A novelty coastal susceptibility assessment method: application to Valdelagrana area (SW Spain),. J. Coast. Conserv. 22, 973–987. https://doi.org/10.1007/s11852-017-0552-2.
Royal Meteorological Society, 2017. Beaufort scale. [Online] Available at. https://www.rmets.org/weather-and-climate/observing/beaufort-scale, Accessed date: 23 October 2017.
Samarakoon, M.B., Tanaka, N., Iimura, K., 2013. Improvement of effectiveness of existing *Casuarina equisetifolia* forests in mitigating tsunami damage. J. Environ. Manag. 114, 105–114.
Schmelz, G.W., Mezich, R.R., 1988. A preliminary investigation of the potential impact of Australian pines on the nesting activities of the loggerhead turtle. In: Schroeder, B.A., compiler) (Eds.), Proceedings of the Eighth Annual Conference on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFSSEFC-214. On File at South Florida Ecosystem Office. US Fish and Wildlife Service, Vero Beach, Florida, pp. 63–66.
Schmid, J., Addison, D., Donnelly, M., Shirley, M., Wibbels, T., 2008. The effect of Australian pine (*Casuarina equisetifolia*) removal on loggerhead sea turtle (*Caretta caretta*) incubation temperatures on keewaydin island, Florida. J. Coast. Res. 55, 214–220.
Sealey, N., 2006. The cycle of casuarina-induced beach erosion - a case study from Andros, Bahamas. In: The 12th Symposium on the Geology of the Bahamas and Other Carbonate Regions, pp. 197–205 (*2004*).
Shanker, K., Pandav, B., Choudhury, B.C., 2004. An assessment of the olive ridley turtle (*Lepidochelys olivacea*) nesting population in Orissa, India. Biol. Conserv. 115, 149–160.
Tanaka, N., 2009. Vegetation bio-shields for tsunami mitigation: review of effectiveness, limitations, construction, and sustainable management. Landsc. Ecol. Eng. 5, 71–79.
Thieler, E.R., Hammer-Klose, E.S., 2000. National assessment of coastal vulnerability to sea-Level rise: preliminary results for the US Pacific coast. Woods Hole. United States Geological Survey (USGS), MA, pp. 1 Open File Report 00-178.
Tinley, K., 1985. Coastal Dunes of South Africa. South African National Scientific Programmes Report No 109 CSIR, pp. 300.
Tsoar, H., 2001. Types of aeolian sand dunes and their formation. In: In: Balmforth, N.J., Provenzale, A. (Eds.), Geomorphological Fluid Mechanics. Lecture Notes in Physics, vol 582. Springer, Berlin, pp. 403–429.
UNDRO, 1982. Natural Disasters and Vulnerability Analysis. Office of the United Nations disaster relief co-ordinator, Geneva.
Vitousek, P.M., 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. Oikos 57, 7–13.
Wallace, B.P., Dimatteo, A.D., Bolten, A.B., Chaloupka, M.Y., Hutchinson, B.J., Abreu-Grobois, F.A., Mortimer, J.A., Seminoff, J.A., Amorocho, D., Bjorndal, K.A., Bourjea, J., 2011. Global conservation priorities for marine turtles. PLoS One 6 (9), e24510. https://doi.org/10.1371/journal.pone.0024510.
Wheeler, G.S., Taylor, G.S., Gaskin, J.F., Purcell, M.F., 2011. Ecology and management of Sheoak (*Casuarina* spp.), an Invader of Coastal Florida, U.S.A. J. Coast. Res. 27, 485–492.
Wood, D.W., Bjorndal, K.A., 2000. Relation of temperature, moisture, salinity, and slope to nest site selection in loggerhead sea turtles. Copeia 2000 119-119.
Zhong, C., Zhang, Y., Chen, Y., Jiang, Q., Chen, Z., Liang, J., Pinyopusarerk, K., Franche, C., Bogusz, D., 2010. *Casuarina* research and applications in China. Symbiosis 50, 107–114.